Waste Management 89 (2019) 354–365

Contents lists available at ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Combustible waste collected at Danish recycling centres: Characterisation, recycling potentials and contribution to environmental savings

Giorgia Faraca *, Vincent M. Edjabou, Alessio Boldrin, Thomas Astrup

Department of Environmental Engineering, Technical University of Denmark, DK-2800 Kgs. Lyngby, Denmark

ARTICLE INFO

Article history:
Received 18 December 2018
Revised 28 February 2019
Accepted 1 April 2019
Available online 16 April 2019

Keywords:
Sampling
Contamination
Circular economy
Recycling rate
Life cycle assessment
Climate change

ABSTRACT

Europe is currently adapting its waste management strategies towards the increased recycling of waste materials, motivated by ambitious recycling targets. This requires correctly sorting and recovering of all relevant waste flows. In Denmark, a considerable share of residential household waste is collected at recycling centres, 16% of which is sent to energy recovery in the form of "small combustible waste". Although essential in order to enhance the management of household waste, very little information exists on its composition. In this study, 25 tonnes of small combustible waste were sampled from eight Danish recycling centres and classified according to material fraction, application and physical properties. On this basis, the potential contribution to the overall recycling rate was evaluated together with estimation of the potential environmental savings associated with recycling of these fractions. Less than half of the sampled waste comprised combustible materials, whereas recyclable fractions accounted for 47–64%, mainly including textiles, plastics and paper waste. Assuming this composition applicable to the national level, recycling these waste materials collected as small combustibles increased national recycling rates for households by 12%, calculated as waste received at recycling processes. Moreover, the potential climate change savings associated with recycling of Danish household waste increased by 30% compared to the current level. Plastics, textiles and paper were the main contributors to this increase, suggesting that improved sorting practices for these materials should be prioritised. The study demonstrates that detailed compositional data for waste materials has paramount importance when estimating recycling potentials and quantifying the associated environmental benefits.

1. Introduction

Waste collection is an important part of modern waste management systems, depending on waste type and quantity, source segregation scheme and legislation, thus influencing the type and quality of materials for downstream management (Nilsson and Christensen, 2011). In general, municipal solid waste is collected from household premises, kerbsides, drop-off points or collection points (recycling centres). Recycling centres (RCs) focus mainly on mono-material collection, as citizens bring their own waste and sort it into about thirty to forty material fractions (only food waste is not accepted). RCs represent a solution for items that are costly or cannot be collected at the kerbside (WRAP, 2016). RCs are a popular collection method in Scandinavian countries, owing to a generally low population density and the abundance of single-family houses; Krook and Eklund (2010) reports that 30% of Swedish household waste is collected at RCs, while in Denmark RCs represent 40–54% of residential household waste and 12–16% of municipal waste (Bisinella et al., 2017a; Natur and Miljø og Fødevareministeriet, 2018; Miljøstyrelsen, 2017, 2018; Statistics Denmark, 2018), thereby covering a substantial portion of the generated waste. While small enterprises are in principle allowed to deliver their waste, their contribution is considered to be low (Miljøstyrelsen, 2018). RCs aim at source-separating all

Abbreviations: CC, climate change; CO_2, carbon dioxide; CR-R, collection for recycling rate; DK, Denmark; EC, European Commission; EoL-RR, end-of-life recycling rate; EU, European Union; FU, functional unit; HDPE, high density polyethylene; LCA, Life Cycle Assessment; LDPE, low density polyethylene; MS, Member State; MSWI, municipal solid waste incinerator; PET, polyethylene terephtalate; PP, polypropylene; PS, polystyrene; PVC, polyvinyl chloride; RC, recycling centre; RR, recycling rate; SC, sensitivity coefficient; SCW, small combustible waste; SR, sensitivity ratio; WEEE, waste electrical and electronic equipment.

* Corresponding author.
E-mail address: gfar@env.dtu.dk (G. Faraca).

https://doi.org/10.1016/j.wasman.2019.04.007
0956-053X/

waste with a recycling potential. In Denmark, items that cannot be recycled are placed in the "small combustible waste" (SCW) container and sent to a municipal solid waste incinerator, while inert materials are placed in a "waste for landfilling" container. SCW made up 16% of the total waste collected at RCs in 2016 (Bisinella et al., 2017a); only 3% was sent to landfill. However, it is estimated that part of the SCW fraction contains misplaced recyclable materials, suggesting that better sorting may increase recycling rates at RCs (WRAP, 2016).

In the last decades, the European Union (EU) has prioritised material recycling, to reduce environmental impacts from production of primary materials as well as supply risk of critical materials (Ardente and Mathieux, 2014). In this context, countries' efforts to improve waste management systems are quantified and monitored through recycling rates. Binding minimum recycling targets exist on several waste fractions, e.g. organic waste, waste electrical and electronic equipment (WEEE) and packaging waste made of selected material fractions, with the aim of fostering resource efficiency and secondary material markets (Reichel et al., 2016). The EU has recently increased the recycling targets for 2025 and 2030 (EU, 2018a, 2018b). In addition, the calculation method for recycling targets was modified: from 2018, recycling rates capture the quantity of waste "which enters the recycling operation" (instead of the quantity of waste "sent for recycling" as it was generically described in Directive 94/62/EC), thus aligning the definition for all European countries and excluding from the indicator all material losses occurring during pre-treatment operations (EU, 2018a). Such ambitious recycling targets will pose challenges even to the best-performing countries (Cimpan et al., 2015a).

Danish waste management has historically focused on incineration with energy recovery. However, in 2013, the Danish government launched the "Denmark without waste" strategy, aiming at increasing recycling for a large number of household waste fractions (The Danish Government, 2013). RCs play a key role in this process, as the presence of trained staff assures good segregation of the fractions that can be sent to recycling (Krook et al., 2008). However, while it is acknowledged that contamination is an issue for recycling (e.g. Eriksen et al., 2018; Faraca et al., 2017; Pivnenko et al., 2015), less attention has been placed on recyclable materials entering waste streams not destined for recycling (Villanueva and Eder, 2014). In this perspective, unrecovered fractions act as obstacles not only to increasing recycling rates, but also to economic growth, as valuable materials are lost (Bourguignon, 2016). At Danish RCs, unrecovered fractions originate when recyclable waste is mis-sorted into other containers, e.g. SCW. However, without very detailed information on the composition of SCW, an estimate of how much recycling potential is lost is not possible.

Besides recycling rates, life cycle assessment (LCA) can be used to evaluate the environmental performance of waste management systems as basis for decision-making. While LCA embraces several impact categories, climate change (CC) often play an important role, because of robust calculation metrics and the fact that climate change is a well-recognised challenge associated with large anthropogenic CO_2 emissions (IPCC, 2013; Steffen et al., 2015). Nevertheless, the actual savings depend to which extent the secondary materials can replace the corresponding virgin ones, which is linked to the quality of the waste materials, in turn influenced by the level of initial sorting achieved at the RCs. The lack of detailed information on the composition of SCW generally prevents an estimation of the loss of recycling potential and the related CC benefits.

Assessing the composition of SCW collected at RCs can be a challenging undertaking, due to the wide range of different fractions delivered daily, including bulky items (WRAP, 2016). There is a significant paucity of data related to the composition of SCW from RCs: DEFRA (2009) estimated it as the most significant data gap in waste composition analysis, together with littering. This in turn reflects a complete lack of estimates on recycling potential that is lost when SCW is directed to incineration. On the other hand, CC savings from recycling individual fractions have been studied widely in the scientific literature. For example, Cimpan et al. (2015b) demonstrated that the high CC savings seen in enhanced separate collection and recycling strategies hold true even for waste management systems dominated by a very efficient waste-to-energy facility. However, Bisinella et al. (2017b) proved that LCA results depend significantly on assumptions concerning waste material composition. Since the physical composition of source segregated waste fractions depends on the method of collection, studies assessing recyclable materials from RCs are crucial for supporting waste management decisions. Environmental impacts of waste collected at RCs have been neglected by the scientific community, despite upstream sorting is generally being considered more efficient than the recovery of recyclables from mixed waste (Villanueva and Eder, 2014).

The aim of this study is to quantify the potential benefits associated with improved sorting of recyclable materials from SCW collected at Danish RCs. This is achieved by: (1) sampling and characterising SCW from eight RCs with respect to the following aspects: material fractions, application type and other characteristics (see Section 2.1.2), (2) assessing the recyclability of the material fractions present in SCW and quantifying the potential contributions to national recycling rates as a result of the correct upstream sorting of these waste materials and (3) evaluating the potential CC savings that could be achieved at the national level by unlocking the recycling potential of recyclable waste currently mis-collected as SCW. The study thereby supports decision-making related to improving the waste management system and potentially achieve regulatory recycling targets.

2. Materials and methods

2.1. Characterisation of small combustible waste

2.1.1. Sampling and sorting campaign

The sampling campaign was carried out between March 2016 and June 2017 and involved eight RCs in the following Danish municipalities: Silkeborg, Lemvig, Roskilde, Hvidovre and Høje Taastrup. The municipalities where the sampling took place have different characteristics: Hvidovre is a municipality (~50,000) included in the urban area of Copenhagen (the capital of Denmark, ~600,000 inhabitants), Roskilde and Høje Taastrup are urban areas (~50,000 inhabitants) in the Sealand region and Silkeborg and Lemvig are located in the Northern Jutland countryside with 50,000 and 20,000 inhabitants, respectively (Appendix B). Whilst the list of recyclable fractions varies according to municipality, the sorting guidelines for SCW waste are very similar and include items with one dimension smaller than 1 m that are not targeted for recycling (ARC, 2018; ARGO, 2018).

The sampling followed the principles set out in Edjabou et al. (2015) and covered ~25 tonnes of waste in total (Table B.1 in Appendix B). For each RC, one container for SCW (corresponding to waste generation of about one week) was unloaded onto the ground. The waste was then sorted manually into individual material fractions according to the classification scheme in Section 2.1.2. The weights of these material fractions were registered to determine overall material composition. Subsamples were then obtained by filling 120L paper bags that were alternatively saved or discarded; saved bags represented one subsample. These samples were stored in ambient conditions until further handling in the lab. Any effects from seasonal variations on waste composition were not investigated in the study.

2.1.2. Composition of small combustible waste

SCW waste was classified according to a tiered level approach (Table 1): (i) material fraction, (ii) application type (i.e. the market sector that a waste product was used in) and (iii) other characteristics (when possible), according to Edjabou et al. (2015). The first classification level (12 fractions) was grouped into targeted or non-targeted for recycling, depending on whether the material fractions could potentially be collected separately at RCs and sent to recycling (see also Appendix C). In this study we defined as "targeted for recycling" those material fractions that are prioritised for recycling as part of the Danish resource strategy (The Danish Government, 2013). For the sake of readability, the terms "recyclable/non-recyclable" will be used to indicate "Targeted/Non-targeted for recycling".

2.2. Recycling indicators

When recyclable waste is collected as SCW, a corresponding recycling potential is lost. To quantify this recycling potential, data on primary household waste generation and treatment were sourced from the Danish Environmental Protection Agency for 2016 (ADS, 2017; Miljøstyrelsen, 2018). The classification of waste treatments includes 13 recovery treatments (R1 to R13; see Table D.1). Only waste flows registered as recycling of organic substances, recycling of metals and recycling of inorganic substances were considered and included herein as "waste sent to pre-treatment". Waste flows referring to "temporary treatment of the waste" prior to recovery (e.g. exchange or storage) were excluded, as no information about their further management was available and because the classification of waste in these categories typically changes after storage (i.e. recycling may not occur). Three recycling indicators (Fig. 1) were calculated for each recyclable waste fraction found in SCW: glass, paper, cardboard, metals, plastic, wood, WEEE and garden waste, excluding textile waste, due to unavailability of primary generation data, as this fraction is often collected by charity organisations (Woolridge et al., 2006). The indicator quantifying waste collected and sent for recycling (CR-R), as expressed in Directive 94/62/EC, was calculated by following Eq. 1. This indicator represents the "traditional" approach for estimating recycling rates in Europe:

$$\mathrm{CR-R} = \frac{\text{Waste sent to pre-treatment}}{\text{Primary waste}} \tag{1}$$

The indicator quantifying waste entering recycling operations (RR), as expressed in EU (2018b), was calculated following Eq. 2:

$$RR = \frac{\text{Waste sent to recycling}}{\text{Primary waste}} \tag{2}$$

Finally, the indicator for the amount of waste converted to secondary materials (end-of-life recycling rate, EoL-RR; Eurometaux, 2012) was calculated following Eq. 3.

$$\mathrm{EoL\text{-}RR} = \frac{\text{Recycled material}}{\text{Primary waste}} \tag{3}$$

In order to calculate RR and EoL-RR, average European sorting and recycling losses were compiled from a review of more than 50 publications (listed in Appendix E). The three recycling indicators were calculated for two scenarios: while the baseline scenario

Table 1
Tiered classification of small combustibles waste (SCW) collected at Danish recycling centres. Recyclable: targeted for recycling; Non-recyclable: non-targeted for recycling.

Material fraction	Application type	Other characteristics
Recyclable fractions		
1 – Plastics	1.1 – Plastic packaging	1.i.1 – HDPE
	1.2 – Other plastics	1.i.2 – LDPE
	1.3 – Plastic foils	1.i.3 – PET
		1.i.4 – PP
		1.i.5 – PVC
		1.i.6 – PS
		1.i.7 – Other polymers
2 – Metals	2.1 – Metal packaging	2.i.1 – Ferrous
	2.2 – Other metals	2.i.2 – Non-ferrous
3 – Glass	3.1 – Glass packaging	
4 – Paper	4.1 – Office, magazines/booklets	
	4.2 – Packaging, wrapping paper	
	4.3 – Other	
5 – Cardboard		
6 – Wood	6.1 – Suitable for recycling (off-cuts, packaging, furniture)	
	6.2 – Suitable for incineration (panelboard, old/treated)	
7 – WEEE[a]	7.1 – Household appliances	
	7.2 – IT and telecommunications	
	7.3 – Consumer equipment + PV	
	7.4 – Lighting equipment	
	7.5 – Toys, leisure and sports equipment	
8 – Textiles[b]	8.1 – Clothes	
	8.2 – Accessories (bags, shoes, etc.)	
	8.3 – House decoration	
9 – Garden waste		
Non-recyclable fractions		
10 – Combustibles	10.1 – Residual household waste	
	10.2 – Other combustibles	
11 – Non combustibles		
12 – Hazardous waste	12.1 – Impregnated wood	
	12.2 – Sprays and containers with hazardous content	
	12.3 – Toner	
	12.4 – Lighter	
	12.5 – Medicines	

[a] Directive 2012/19/EU-Annex I covers 10 WEEE categories. Only detected categories are inserted in this table.
[b] According to Nørup et al. (2018).

Fig. 1. Definition of recycling indicators with respect to recycling treatment phases. CR-R: collected for recycling rate; RR: recycling rate; EoL-RR: end-of-life recycling rate.

describes the current waste management system in Denmark, the SCW scenario represents the case in which the recyclable fractions contained in SCW are sorted correctly and sent to recycling. In scenario SCW, the recycling potential from recyclable fractions currently collected as SCW was obtained by applying SCW material composition results (Section 3.1) to the total amount of SCW waste collected at Danish RCs in 2016, which amounted to 263,000 tonnes. In both scenarios, recycling indicators were calculated for total recyclable waste generated by households.

2.3. Life cycle assessments

2.3.1. Goal, scope and scenario definition

The goal of the assessment was to calculate the potential environmental savings provided by improved sorting of SCW. This was done by comparing impacts from recycling the recyclable fractions in SCW (i.e. the "recycling scenario") with impacts from the current incineration of the same fractions (i.e. the "incineration scenario"). The functional unit (FU) was the management of the yearly amount of recyclable waste mis-collected as SCW at Danish RCs (i.e. 144,845 tonnes of recyclable waste; composition provided in Appendix H). The goal was achieved by subdividing the assessment into nine reference flows, with each flow representing one tonne of each of the nine recyclable fractions under study: glass, paper, cardboard, metals, plastic, wood, WEEE, textiles and garden waste. Results for the individual fractions were then scaled to the national level (i.e. the FU). The study followed ISO guidelines and adopted a consequential approach to reflect the consequences of a change in waste management (ISO, 2006). The temporal scope was 2018–2050. The geographical scope was Denmark; however, for those recyclable fractions that are currently entirely exported to other countries for recycling, typical European solutions and efficiencies for recycling of waste were assumed (in the absence of more detailed information).

The system boundaries start after the collection of the waste material fractions at RCs and end at the point of substitution of secondary products/energy and/or final management of any residual fractions (see the case of paper in Fig. 2, other scenarios in Appendix F). This excludes the collection phase at the RCs; being it identical for the two scenarios, the exclusion has no consequences for the comparison. The incineration scenario assumed the treatment of the fractions targeted for recycling in a municipal solid waste

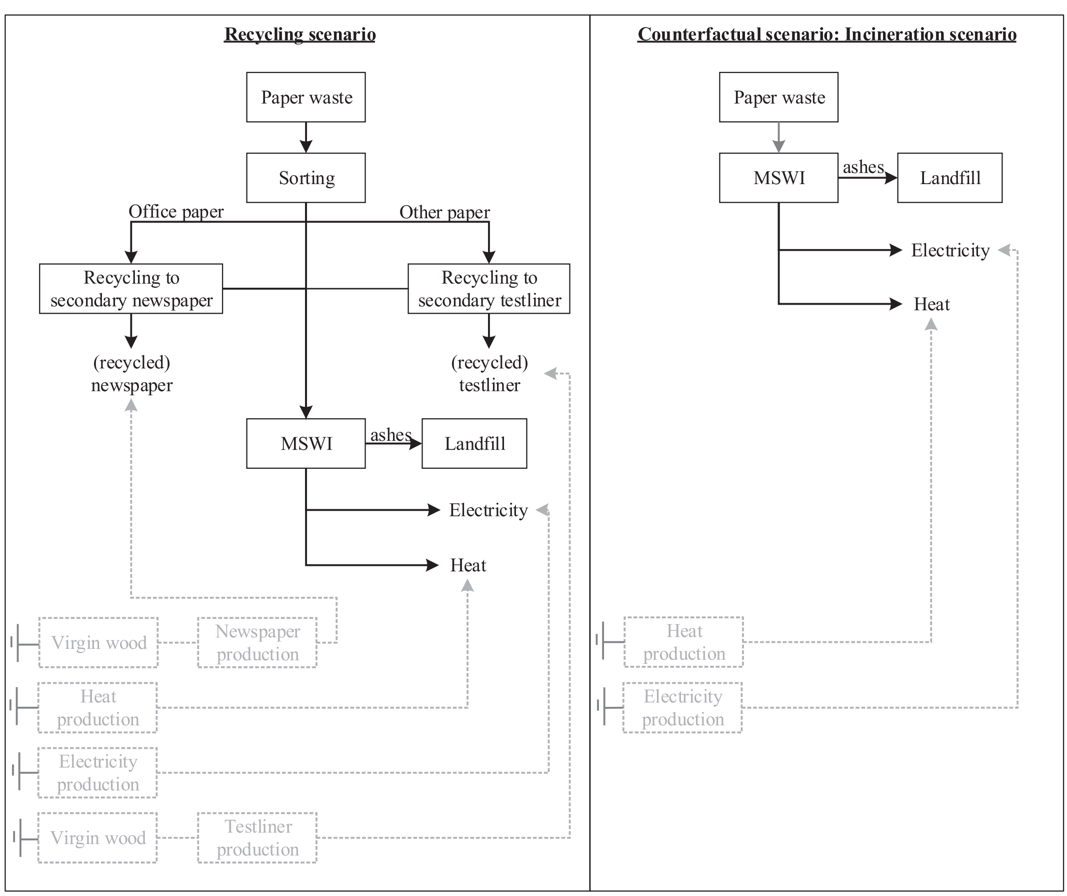

Fig. 2. System boundaries and scenario definition for the case of paper waste as an example. Diagrams for the other material fractions can be found in Appendix F.

incinerator (MSWI) with electricity and heat recovery (22% and 73% efficiency respectively; Møller et al., 2013). This incineration scenario was considered as the *counterfactual* scenario, meaning the scenario which is avoided by the change in the system (i.e. sending recyclable fractions to recycling). In the recycling scenarios, material sorting was assumed to occur upstream, i.e. by citizens prior to collection. The recycling scenarios are described in the following:

- Recycling of glass. After sorting out impurities, the waste glass is converted to cullets, which are fed into the melting furnace for production of secondary glass. Waste glass needs the addition of virgin materials (minimum 17%) to ensure a satisfactory outcome (Larsen et al., 2009).
- Recycling of paper. A sorting facility separates waste office paper from the rest; office paper is then shredded, separated from contaminants, pulped, bleached and deinked. The deinked pulp is recycled into newsprint (Haupt et al., 2018; Turner et al., 2015). Paper packaging and other paper are shredded, separated from contaminants, pulped and deinked. The deinked pulp is recycled into testliner (the outer part of corrugated cardboard).
- Recycling of cardboard. Waste cardboard is shredded, separated from contaminants, pulped and deinked. The deinked pulp is recycled into testliner.
- Recycling of metals. Metals are sorted mechanically into ferrous and non-ferrous metals. Ferrous scraps are smelted in an electric arc furnace and cast into secondary steel foils. Non-ferrous scraps undergo mechanical shredding and magnetic, sink-and-float and eddy current separation, to isolate aluminium scraps that are melted, alloyed and cast into secondary aluminium ingots.
- Recycling of plastic. Plastic waste enters a plastic sorting facility, where plastic films are sorted by ballistic separation and recycled into a polyolefin mix for the production of plastic profiles (Rigamonti et al., 2014). Hard plastic is sorted into major polymers (PP, HDPE, PVC, PET and PS), each entering a homopolymer reprocessing line to produce secondary granulates, which are extruded to non-food packaging materials.
- Recycling of wood. Wood waste suitable for recycling is shredded and dried. The dried chips are then pressed, with the addition of organic resins, to form a solid mat (particleboard). Secondary particleboard in Denmark is produced with the addition of virgin materials (around 20%) to ensure a satisfactory outcome (Vis et al., 2016).
- Recycling of WEEE. WEEE enters a sorting facility, where it is shredded and sorted into individual material fractions. Plastic, glass and ferrous and non-ferrous metals are sent to recycling (see paragraphs corresponding to the individual fractions). Batteries are sorted further and sent to a conversion facility that recycles lead and cobalt for use in secondary batteries. Other precious metals (e.g. gold, silver, platinum, etc.) are extracted from mobile phones. Glass from cathode ray tubes (CRTs) is sent to a pyro-metallurgical process that recovers lead used in secondary batteries, while copper from cables and CRTs is recovered for secondary cathodes.
- Recycling of textiles. Textiles are sorted manually into clothes, accessories (shoes, bags, etc.) and house textiles (carpets, pillows, etc.). Clothes are sorted manually into natural and synthetic fibres, which are then physically freed from impurities, cut to a specified size and upgraded to secondary wipers (Bartlett et al., 2013). Accessories are mechanically shredded, sorted from contaminants (magnetic and cyclone separation) and entwined before being mechanically cut into secondary matrass foam filling. House textiles are mechanically shredded, sorted from contaminants and reprocessed into composite surfaces (Turner et al., 2015).
- Composting of garden waste. Garden waste is assumed composted into open windrows and used for private gardening purposes (Boldrin et al., 2011).

Environmental impacts were calculated for all impact categories contained in the ILCD recommended method (see Appendix G; EC-JRC, 2011). No normalisation nor weighting was applied. In the main article, the interpretation of the results focuses on Climate Change; detailed results for all impact categories can be found in Appendix M. The computation was performed with the EASETECH software (for details, see Clavreul et al., 2014).

Multi-functionality of processes was solved by substitution by system expansion as recommended for consequential LCAs; this implies identifying the marginal suppliers (Weidema et al., 2009). For the provision (and substitution) of electricity, the marginal source for Denmark was assumed to consist of 56% wind energy, 28% biomass-energy and 16% natural gas, according to Munõz (2015), as these are the sources expected to increase their capacity by 2050 (EC, 2016) and thus the most competitive ones (Weidema et al., 2009; see Appendix K.1). The marginal source for the provision (and substitution) of heat was assumed to consist of natural gas, constrained by the infrastructure related to district heating (Fruergaard et al., 2010). The marginal product for recycled materials was assumed to be the corresponding products from virgin sources, according to Haupt et al. (2018) and ecoinvent (2018) data. In the case of textile recycling, recycled wipers were assumed to substitute for wipers made of virgin paper or PP for natural or synthetic fibre clothes, respectively; secondary matrass filling from recycled textile accessories was assumed to substitute for polyurethane foam (Turner et al., 2015). Compost from garden waste was assumed to substitute for mineral fertilisers (potassium chloride, diammonium phosphate and urea ammonium nitrate) and peat (Boldrin et al., 2010; Tonini et al., 2015). Any by-products (residues) originating from the waste reprocessing stage were assumed to be incinerated in a MSWI with electricity and heat recovery. Any metal scraps recovered from the MSWI were assumed to be recycled; fly ashes were assumed to be disposed of in dedicated landfill, while bottom ashes were assumed reused in road construction, as gravel substitute.

It has been suggested that for some waste materials secondary production has increased dramatically to the point that material supply for primary production has been affected to also include a share of secondary materials. For example, in the case of glass and steel primary production, the ecoinvent database (2018) includes the addition of 84% glass cullet and 20% iron scrap, respectively. In this study, recycled products were assumed to substitute for virgin materials, due to the fact that a residue cannot represent the marginal alternative, as it is not able to adapt its production in response to a system change (Weidema et al., 2009).

2.3.2. Life cycle inventory

Data used for the modelling were compiled from an extensive review of scientific articles, reports and databases (see the complete list of references in Appendix I). Such data were inserted into the model as parameters defined by a probability distribution in order to include related uncertainty (see Section 2.3.3). Data on the foreground system (including waste management activities) were sourced mainly from Andreasi Bassi et al. (2017), Biganzoli et al. (2015), ecoinvent (2018), Haupt et al. (2018), Rigamonti et al. (2014) and Turner et al. (2015). Data on the background system (interacting with the foreground system by supplying energy

and material inputs) were sourced mainly from ecoinvent (2018). However, ecoinvent datasets were edited to adapt the energy sources to Danish conditions. When the ecoinvent datasets represented processes assumed to occur outside Denmark (i.e. for steel, aluminium, copper and plastic primary production, according to Cimpan et al., 2015b), the database was not edited. Baseline values for pre-treatment losses and recycling losses reflect those used in Section 2.2. Capital goods were included in the assessment and assumed to adjust in capacity according to changes in demand (in line with a long-term temporal scope; Weidema et al., 2009).

2.3.3. Uncertainty, sensitivity and scenario analysis

Making use of the parameters' probability distributions, uncertainties associated with each input parameter were propagated to calculate the uncertainty associated with the results (Bisinella et al., 2016). Based on Monte Carlo analysis (1,000 simulations), input values were sampled from the probability distributions, and related results were calculated. The obtained minimum and maximum results provided the uncertainty range of the results, assuming triangular data distribution.

In order to identify how results vary as a consequence of a change in input values, a sensitivity analysis was carried out. Each parameter was increased by 10% of its value in a "one-at-a-time" manner while keeping all other parameters fixed at their baseline values (perturbation analysis). The results enable the calculation of sensitivity ratios (SRs) associated with each parameter (Equation J.1 in Appendix I), providing information about the sensitivity of the model to each parameter.

Key assumptions associated with the background system (interacting with the foreground system by supplying energy and material inputs, including avoided energy and virgin material production) and their influence on the results were evaluated by scenario analysis on: i) type of electricity provision and ii) assessment method used for biogenic carbon emissions (in terms of CC only). In the first case, the baseline assumption of marginal electricity production from wind, biomass and natural gas was replaced by a fossil alternative of 98% coal and 2% natural gas (see Appendix K.1). Similarly, the baseline accounting method for biogenic carbon dioxide emissions (which are generally considered climate neutral) was compared to a recent methodology which assumes that the impact of biogenic CO_2 emissions depends on the rotation period of the wood under assessment (Cherubini et al., 2011, 2016; Faraca et al., 2019; see Appendix K.2)

3. Results and discussion

3.1. Composition of small combustible waste

The material composition (first level) of SCW sampled at eight Danish RCs is shown in Fig. 3. "Combustibles" was the largest category, representing on average 38% of the sampled waste, ranging from a minimum of 24% to a maximum of 47%. Other categories making a significant contribution were: Textiles (average: 13%), Plastic (12%), Paper (10%), Wood (9%), Non-combustibles (7%) and Cardboard (5%). Finally, Metals (average: 2%), WEEE (2%), Garden waste (2%), Hazardous waste (2%) and Glass (0%) contributed very little to the total waste. The composition was in accordance with data for the UK (Resource Futures, 2011), where major shares of contamination in SCW were reported in terms of wood (15%), non-combustibles (15%), plastic (11%), paper (5%) and textiles (5%). Krook and Eklund (2010) also reported the large presence of paper and wood in SCW (20% of the total SCW in our study). The obtained results suggest that there are large opportunities for enhancing the collection and sorting SCW. Recyclables represented 47–64% of SCW, which constitutes a potential for recycling which is lost when SCW is incinerated.

The composition of SCW appeared to differ considerably between RCs: the standard deviation for the first-level categories varied between 21% and 95% of the corresponding average value (Table L.1). This indicates that the composition of SCW may vary both with time and location, and that elaborate waste characterisation efforts may be needed to achieve robust composition data at a local level. While the true variability in composition data for SCW is not available at the national level in Denmark, the detailed sampling of the eight RCs discussed herein may comprise a considerable portion of such variability, due to the variety of locations included in the study. As such, applying the median values (dark-

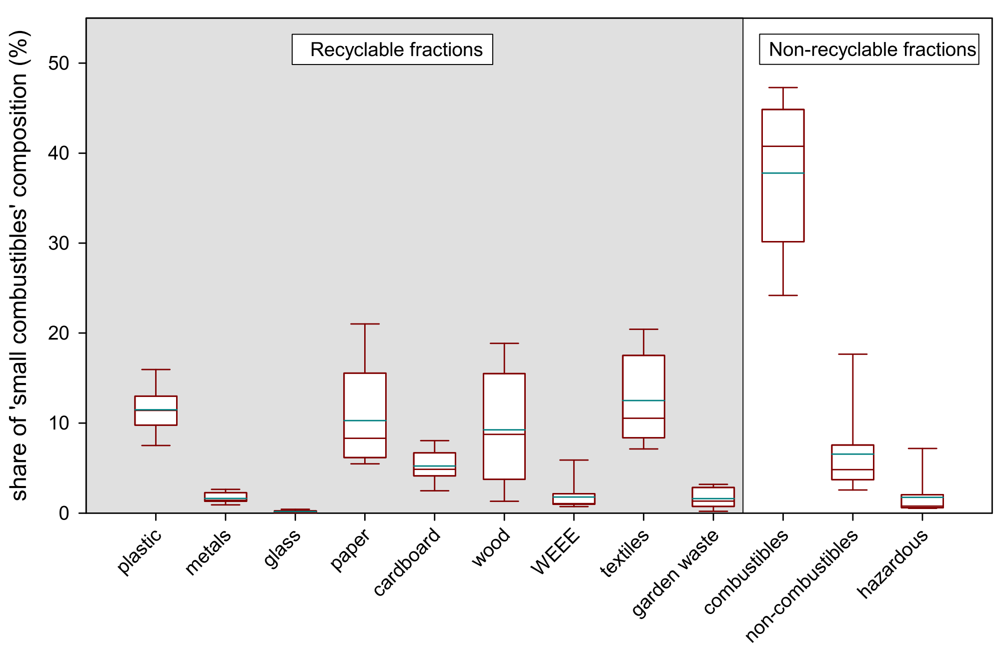

Fig. 3. Material composition of the small combustible waste (SCW) fraction collected at recycling centres (RCs) according to Table 2 (first level). The dark-red lines inside the boxplots represent the mode, the dark-cyan the median.

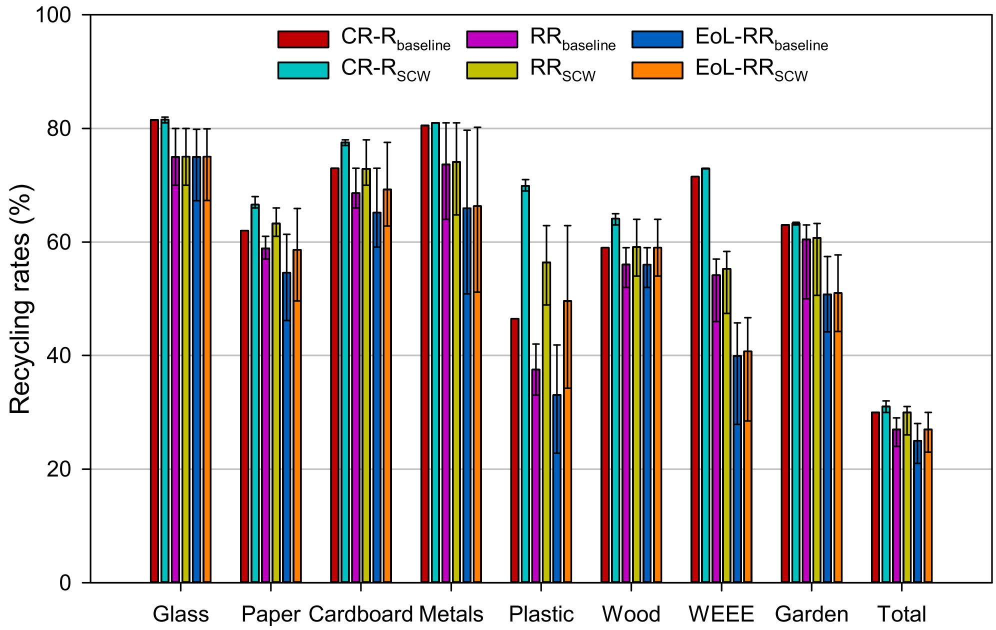

Fig. 4. Recycling rates CR-R (collected for recycling rate), RR (recycling rate) and EoL-RR (end-of-life recycling rate) for different recyclable fractions. The baseline scenario assumes Recyclable fractions from SCW sent to incineration. The SCW scenario assumes improved collection sorting, redirecting Recyclable fractions in SCW to recycling activities. Textile waste is not included, due to the lack of data on primary generation. Error bars display the min and max values obtained by using the entire distributions for the parameters representing sorting losses (listed in Appendix E).

cyan line in Fig. 3) for further modelling was considered reasonable, especially in the absence of other data. European studies on municipal waste composition report little, generic or no information on result variations, but they do identify differences in source segregation schemes as possible reasons, along with sorting guides, income levels and demographics (DEFRA, 2009; Edjabou et al., 2015).

Table 2 summarises the results for the application category of each material fraction. This information is important because the application category of recyclable waste may be associated with its quality (Eriksen et al., 2018), since different applications of recyclable fractions have diverse physical and chemical properties – and thus recycling potential. For example, it is known that hard plastic packaging is highly valuable for recycling, whereas plastic films constitute unwanted items at plastic reprocessing facilities (Horodytska et al., 2018); nevertheless, plastic packaging constituted on average only 11% of SCW, while plastic films accounted for 31%. Similarly, information on the composition of textile waste is essential for evaluating environmental impacts as a result of its management, as the recycling process differs across textile application categories (see Section 2.1.2).

3.2. Recycling indicators

Recycling indicators calculated for the recyclable fractions occurring in SCW (excluding textiles) are illustrated in Fig. 4. In the baseline scenario, the highest CR-R$_{baseline}$ indicators were found for glass and metals (above 80%), followed by cardboard and WEEE (73% and 72%, respectively); plastic showed the lowest value (47%). In scenario SCW, the material fractions for which the increase in CR-R was larger compared to the baseline scenario were plastic (+23%), paper, cardboard and wood (+5% respectively). The results owed to the relevance of the quantity of these materials fractions sorted from SCW compared to the total amount collected from households in Denmark: +12%, 13% and 17% for separate collection of paper, wood and cardboard, and +44% for plastic. This is an important finding considering that plastic management is very high on the EU agenda (EC, 2018c). Conversely, the CR-R$_{SCW}$ indicator did not change for the remaining material fractions (for example, glass and garden waste would contribute to 0.3% and 0.7% of the total collection). The obtained percentages indicate that plastic, and to a less extent wood, paper and cardboard, could be prioritised in order to raise recycling targets. The total CR-R$_{baseline}$ was 30%, which is lower than the figures reported in current national waste statistics (48%; Miljøstyrelsen, 2018); however, the CR-R$_{baseline}$ herein does not include gypsum, tyres and organic waste, which represent heavy waste streams and thus a large contribution to the overall recycling rate. The total CR-R$_{SCW}$ increased to 31%, which indicates that the contribution of better SCW sorting to the recycling rate is not to be neglected, especially in view of EU ambitious recycling rates.

Regardless of the scenario, the RR indicator decreased for all recyclable fractions when compared to CR-R. The extent of the decrease was largest for plastic and WEEE, due to the complexity of disassembling the products and the presence of contaminating fractions in the collected waste that are expected to be sorted out during pre-treatment. Also, the ranking of recyclable fractions did change: the indicators for WEEE moved from being in the highest range (as CR-R) to being the third-to-lowest score (as RR). As shown herein, pre-treatment losses can be large, and the RR indicator allowed for a more consistent comparison across waste fractions. However, the obtained results showed that plastic, paper, wood and cardboard still represent the material fractions worth of prioritisation. The total RR$_{baseline}$ was 27%, which increased to 30% in scenario SCW.

The material fractions showing the largest decrease from RR to EoL-RR were WEEE (-26% of RR) and plastic (-15%), whereas material fractions such as glass and wood were not affected (100% recycling efficiency). The EoL-RR reflects recycling process yields and provides information on the actual amount of waste replacing primary materials. Although not included in the recycling targets, the EoL-RR can provide a measure of resource efficiency, if coupled

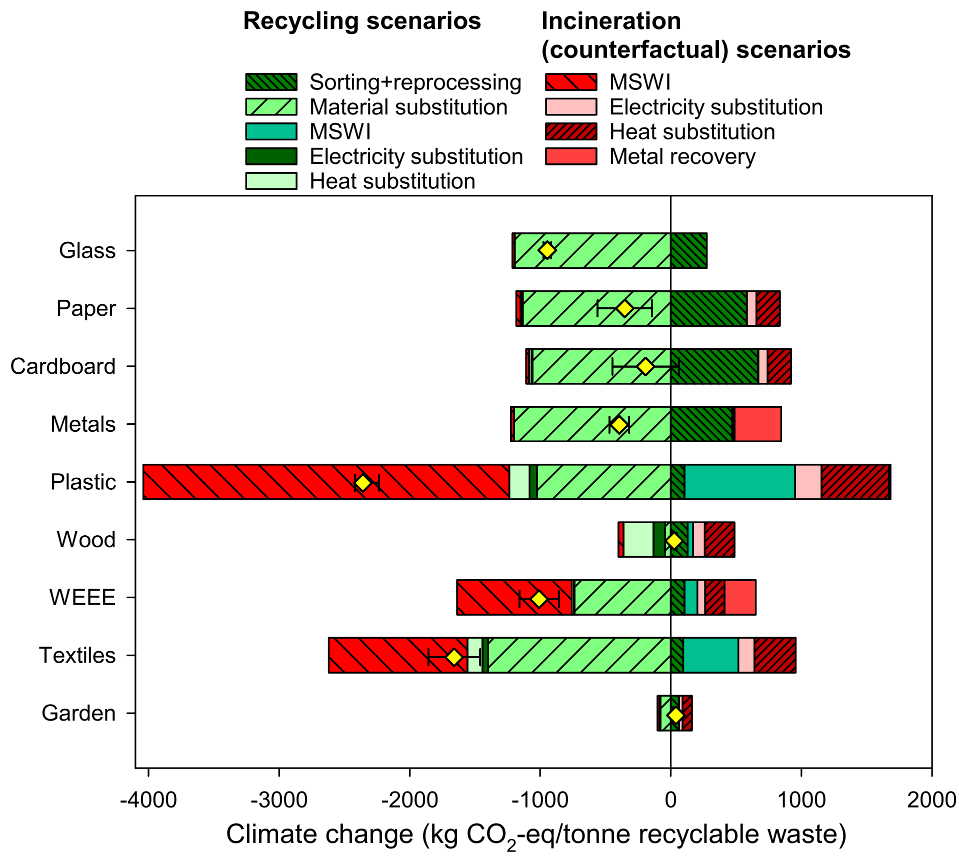

Fig. 5. Climate change results (kg CO_2-eq/tonne recyclable waste) for the treatment of the nine recyclable fractions. Green bars represent the saving/impacts of recycling one tonne of the corresponding material. Red bars represent the avoided savings/impacts of incinerating one tonne of the corresponding material (being the counterfactual scenario, the incineration results were subtracted from the recycling results). Yellow diamonds indicate the net results (recycling scenario minus incineration scenario); error bars represent the standard deviation around the mean value from the Monte Carlo analysis.

with information on the actual recycled content in market products. However, the outcome of the EoL-RR depends on the type of recycling process; for some materials (e.g. plastic or metals), many types of recycling are possible.

3.3. Life cycle assessment: Individual material fractions

The LCA results for the management of the nine recyclable fractions from SCW are displayed in Fig. 5; as the counterfactual alternative, the results of the "incineration" scenario (the red-coloured bars) have been subtracted from the recycling results (the green-coloured bars). By subtracting the incineration scenario we make explicit the net impacts/savings provided by the change in the system (i.e. the recycling scenario) with respect to the baseline scenario. The results indicated CC savings for all material fractions except for wood and garden waste; for these two waste fractions, the results showed a modest burden (31 and 119 kg CO_2-eq/tonne of wood or garden waste, respectively). The largest savings were achieved for recycling plastic, textiles, WEEE and glass as alternative management to incineration; paper, cardboard and metals provided relatively lower CC savings.

Substituting primary materials (in the recycling scenarios) contributed significantly to total savings for most material fractions, especially for glass, paper, cardboard and metal (95–98% of total savings). In the case of plastic, WEEE and textiles, the contribution made by substituting primary materials occurring in the recycling scenarios represented 25%, 44% and 53% of total savings, respectively, the largest contribution being the (avoided) direct emissions from the MSWI in the incineration scenario (69%, 53% and 41%, respectively). Results for wood waste were driven by heat substitution in the recycling scenario (49% of total savings), while those for garden waste by (avoided) direct emissions from MSWI in the incineration scenario (49%). Given the LCA outcomes, it is possible to group the material fractions into three classes: (1) glass, paper, cardboard and metals, providing net savings, for which results were driven mainly by savings brought by recycling, the counterfactual scenario contributing to a minor extent; (2) plastic, WEEE and textiles, providing large net savings, for which a major part of the results was represented by avoided direct emissions in the counterfactual scenario (i.e. incineration), mainly because a large part of the FU is of fossil origin (cfr. material composition in Appendix H); (3) wood and garden waste, for which the counterfactual scenario provided larger savings than recycling, mainly because of the low savings from avoided material production. While for the first group (glass, paper, cardboard and metals) recycling is always beneficial, for the second group (plastic, WEEE and textiles) recycling is *more* beneficial when it avoids incineration, thereby suggesting that primarily these fractions should be sourced from SCW and redirected to recycling. It is essential to note that for WEEE, plastic and textiles, recycling practices are not well established, due to economic and technological barriers, high fragmentation of the recycling chain, material contamination, low sorting efficiency and material complexity in the products (Bourguignon, 2016; Dahlbo et al., 2017; EU, 2018c; Villanueva and Eder, 2014). For example, only 30% of plastic waste was recycled in 2014 in the EU, and recycled plastics only accounted for 4–6% of the EU demand for new plastic materials (EU, 2018c); WEEE recycling relies entirely on manual work, due to the high complexity in the product design, which is not tailored to facilitate recycling (EU, 2018a); according to WRAP (2012), less than 1% of clothing is recy-

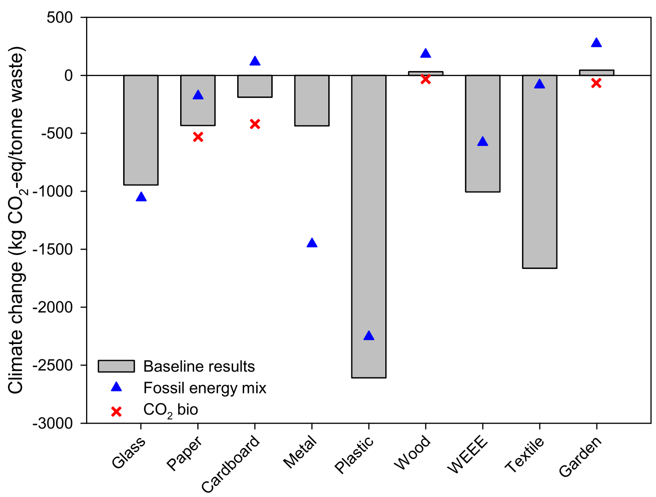

Fig. 6. Scenario analysis on the provision of energy (baseline case: wind, biomass and natural gas; alternative case, indicated by blue-triangular marks: coal) and on the accounting method for biogenic CO_2 emissions (baseline case: biogenic CO_2 emissions considered climate neutral; alternative case, indicated by red crosses: biogenic CO_2 emissions accounted for). Grey-filled bars represent net results from the baseline case (as illustrated in Fig. 4).

cled in the EU, while 56% of used textiles are estimated to be incinerated in Denmark (Schmidt et al., 2016). Large technological and legislative improvements are needed to enhance the recycling of these recyclable material fractions. In addition, WEEE recycling possibly provides savings in terms of resource efficiency: according to the model, for every 1,000 kg of WEEE sent to recycling, 107 kg of precious metals are kept in the material loop, including critical raw materials (like cobalt, indium, palladium and phosphorous) and highly valuable materials (like gold and silver), thereby contributing to security of supply.

It should be noted that while results for the recycling scenario act as stand-alone figures, the net results illustrated in Fig. 5 depend on the counterfactual scenario, which in this case was incineration with electricity and heat recovery; in other countries, the counterfactual scenario could be landfilling of the waste (e.g. in the UK). The results from the recycling scenario were within the range of other findings from the scientific literature (see Table M.1) despite the wide range of results, most of all for paper and cardboard. However, it should be clarified that the studies differed in their goals, geographical scope and type of energy provision, which may greatly affect final results. Moreover, the reviewed studies rarely reported the detailed composition of the waste, therefore making comparisons difficult to interpret.

Results for other impact categories (Appendix N) generally confirmed the outcomes discussed in terms of CC, i.e. savings for all material fractions except for wood and garden waste. For these material fractions, incineration appeared to be a more sustainable alternative, mainly because of the large savings made from substituting marginal heat. Among the other material fractions, no clear conclusion could be drawn on which material to prioritise; performances varied with impact categories, and were driven mainly by the type and amount of material used for reprocessing waste and/or the production of (avoided) virgin materials. Avoided loads from incinerating plastics, WEEE and textiles were not confirmed as important contributors to environmental savings, except for plastic in terms of ecotoxicity.

3.4. Uncertainty and sensitivity analysis

The variation in the results caused by the uncertainty in the input data can be seen in the error bars in Fig. 5. Such variation was relatively large for paper, cardboard, WEEE and textiles waste fractions. This is due to the fact that secondary data from literature showed a wide value range. Conversely, results for glass, metals, plastic, wood and garden waste appeared more robust, due to a narrower data range for the parameters.

The results from the sensitivity analysis showed that in the recycling scenarios, the most sensitive parameters were the sorting losses, recycling losses and substitution factors, irrespective of the waste fraction addressed in the scenario (Appendix O). For paper and cardboard, the parameters related to the quantity of CO_2 fossil emissions released during recycling processes were also significant. The model appeared overall less sensitive to parameters related to energy consumption during reprocessing. Conversely, in the incineration (counterfactual) scenarios, energy conversion efficiencies at power plants were clearly the most important parameters.

3.5. Scenario analysis

Fig. 6 illustrates the potential importance of the assumptions for the electricity mix and the accounting method for biogenic carbon dioxide emissions. Assuming a fossil-based energy mix for the provision of electricity greatly influenced the LCA results. In general, total savings decreased due to avoided larger savings from substituting heat in the MSWI in the incineration (counterfactual) scenario, although direct emissions from energy use during recycling also contributed to the results. The largest decrease was noted for textiles, cardboard and garden waste. On the other hand, total savings increased for metal and glass waste, owing to larger savings from virgin material substitution occurring in the recycling scenario. In the case of a coal-based energy mix, sorting efforts should be directed toward plastic, metals and glass. However, this is unlikely given the ambitious targets for renewable energy to which Denmark is committed. Nevertheless, the results indicate that country-specific energy mixes should be used in the modelling, as they greatly influence the results.

When accounting for impacts from biogenic CO_2 emissions, the results provided increased savings for organic material recycling, owing to an increase in direct emissions avoided from incinerating the waste (in the counterfactual scenario). However, in the case of wood waste, the difference in results is modest compared to the initial assumption (biogenic CO_2 considered as carbon neutral), since direct biogenic CO_2 emissions occur also in the recycling scenario (around 30% of wood waste is incinerated, as it is deemed unsuitable for recycling). Despite the modest difference, the results changed from burdens to savings.

3.6. Life cycle assessment: Combined results

Fig. 7 illustrates the additional yearly environmental impacts/savings provided by the FU, in the case recyclable waste was sorted out from SCW and redirected to recycling (“CC_{SCW}”, dark-cyan bars: the incineration scenario was subtracted from the recycling scenario). These were compared to the current situation, i.e. the yearly savings achieved in Denmark from existing recycling of the assessed nine recyclable fractions (“$CC_{current}$”, dark-red bars: only recycling scenario, no counterfactual management assumed). In total, about 150,000 tonnes of CO_2-eq could be saved annually in Denmark, i.e. 27 kg CO_2-eq/capita/year, upon correct upstream sorting of recyclables. Summing these savings with those achieved yearly in Denmark would represent an increase of 30% compared to the baseline situation, demonstrating that SCW holds great potential for reducing climate change emissions. Due to the important role that waste composition plays in this study, the results cannot be extended to other countries. Moreover, the evolution of the composition of the FU within the time scope was not consid-

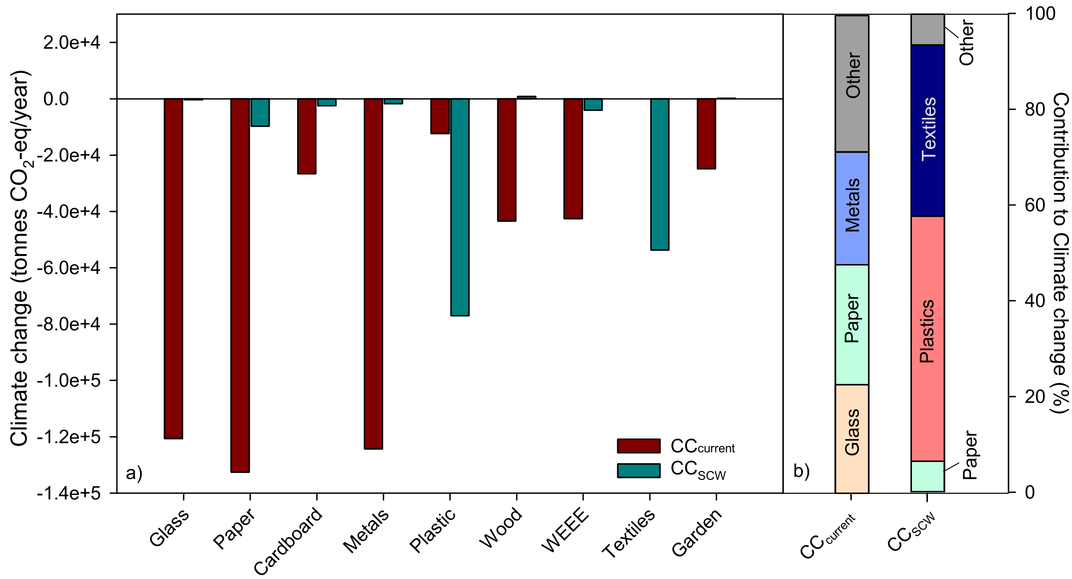

Fig. 7. a) Climate change results (tonnes CO_2-eq/year) saved/released in Denmark from recycling individual recyclable fractions sourced from SCW collected at recycling centres ($CC_{current}$, dark-red filled bars) compared to the additional savings/impacts obtained by correctly sorting and recycling recyclable fractions from small combustible waste at recycling centres in Denmark (CC_{SCW}, dark-cyan filled bars). b) Contribution (%) of major recyclable fractions to climate change.

ered, due to the high uncertainty related to model such changes which may depend on income level, social behaviour, product trends and so on. We strongly recommend that compositional studies should be carried out in order to provide the basis for future compositional forecast.

Recycling paper, plastic and textiles alone represented 96% of annual savings from SCW (dark-cyan bars), highlighting that these material fractions should be the ones on which better sorting practices should focus. Although the sorting of other recyclable fractions would provide additional savings, the environmental gain does not justify the effort. Better sorting practices concentrating on redirecting paper, plastic and textile waste to the right containers at RCs could be ensured by clear signs being put next to the SCW container, and informative guidelines being sent to households. An alternative (but costly) solution could be to introduce a sorting facility step aiming at separating the targeted fractions not only from SCW, but also from mixed waste. The possibilities and role of sorting mixed waste is being evaluated in some countries such as The Netherlands and Austria, while in Spain, France and Greece this option is already in practice, separating an additional 18% of recyclables (Cimpan et al., 2015a). However, it needs to be considered that the mechanical separation of paper, plastic and textiles is not easy, due to the physical properties of the waste fractions; therefore, the work of hand-pickers should complement the presence of machinery (e.g. optical sorters, wind sifters, etc.). It is important to note that while the recycling of paper is established in the EU, plastics and textiles are complex material fractions for which barriers to recycling exist. Technology innovation should focus on increasing the level of recycling process maturity for such material fractions. Solid data on material composition are required for correct process design. It is also important to underline that sorting and recycling losses for these materials are significant, most of all in the case of plastic and textiles (see Fig. 4 and Appendix I): these losses were included in the dark-cyan bars in Fig. 7, suggesting that savings could be even greater in the case of improvements in pre-treatment and/or recycling processes. Finally, the lack of data on textile collection appears a challenge that needs to be tackled due to the large savings potentially provided.

Recycling targets and CC impacts act today as major driving forces for decision making, where recycling targets favour large and heavy waste streams, while CC provides information on potential environmental savings *per unit mass*. This study demonstrates that both indicators should be complemented to obtain a clear picture of the situation and to support decision making transparently. The use of RR indicator appeared to provide information which is more complete than CR-R, as pre-treatment losses are taken into account, and more robust than EoL-RR, which depends on the recycling application. The leaking of valuable resources to MSWI appears to be a cross-cutting issue, involving security of supply and environmental protection. Detailed information on the composition of SCW is paramount to improve current waste management.

4. Conclusions

Small combustible waste (SCW) collected at recycling centres (RCs) was sampled at eight Danish municipalities and sorted manually into 12 detailed material fractions. The composition of SCW was largely contaminated by plastic, paper, wood and textiles. On average, 56% of SCW was identified as recyclable waste, while 32% was combustibles. The results indicated that SCW holds considerable potential for improved management. The consequences of correctly sorting recyclable waste were investigated by addressing the contribution to recycling rates and climate change (CC) savings. If all recyclables in SCW (currently sent to incineration) were redirected to recycling, the recycling rates (as RR, the ratio between waste sent to recycling and primary waste) could be increased by 12%. With the exception of wood and garden waste, all waste fractions in SCW provided CC savings upon recycling. In particular, plastic, WEEE and textile waste provided the largest environmental savings (−2600, −1000 and −1700 kg CO_2-eq/tonne of recyclable waste). When comparing with the current environmental savings provided by recycling of the same waste fractions at national level, however, plastic, textile and paper waste should receive priority as these fractions contributed with 96% of the additional potential of 150,000 tonnes CO_2-eq (i.e. 27 kg CO_2-eq/capita/year) that could be saved annually by unlocking the recycling potential of SCW at Danish recycling centres. Environmental savings currently achieved in Denmark as a result of household waste recycling could be increased by 30% as a result. This high-

lights the important contribution that effective source-segregation of household waste materials may offer for recycling and the associated climate impacts of waste management.

Acknowledgements

The Danish Environmental Protection Agency is highly acknowledged for funding this study. Moreover, the authors are grateful to Econet A/S for providing the opportunity to sample, and to S.A. Bassi (Technical University of Denmark) for useful comments on the study.

Funding

This work was supported by the Danish Environmental Protection Agency (Miljøstyrelsen).

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2019.04.007.

References

ADS (affaldsdatasystemet), 2017. Affaldsproduktion i Danmark fordelt på type og behandling (R013) (Waste generation in Denmark by type and treatment). Available at: (in Danish) Last accessed: 17.09.2018 <https://mst.dk/affald-jord/affald/affaldsdatasystemet/find-affaldsdata/>.

Andreasi Bassi, S., Christensen, T.H., Damgaard, A., 2017. Environmental performance of household waste management in Europe - An example of 7 countries. Waste Manag. 69, 545–557. https://doi.org/10.1016/j.wasman.2017.07.042.

Amager Ressource Center (ARC), 2018. Sadån sorteret du. (Here is how you sort). In Danish. Availabel at: <https://www.a-r-c.dk/media/439682/1652-arc-sorteringsvejledning_web.pdf>.

ARGO, 2018. Sorteringsvejledning. (Sorting guide). In Danish. Available at: <https://argo.dk/privat/sorteringsvejledning/>.

Ardente, F., Mathieux, F., 2014. Identification and assessment of product's measures to improve resource efficiency: the case-study of an energy using product. J. Clean. Prod. 83, 126–141. https://doi.org/10.1016/j.jclepro.2014.07.058.

Bartlett, C., McGill, I., Willis, P., 2013. Textiles flow and market development opportunities in the UK. Waste & Resources Action Programme, Banbury, UK.

Biganzoli, L., Falbo, A., Forte, F., Grosso, M., Rigamonti, L., 2015. Mass balance and life cycle assessment of the waste electrical and electronic equipment management system implemented in Lombardia Region (Italy). Sci. Total Environ. 524–525, 361–375. https://doi.org/10.1016/j.scitotenv.2015.04.041.

Bisinella, V., Conradsen, K., Christensen, T.H., Astrup, T.F., 2016. A global approach for sparse representation of uncertainty in Life cycle assessments of waste management systems. Int. J. Life Cycle Assess. 21, 378–394. https://doi.org/10.1007/s11367-015-1014-4.

Bisinella, V., Brogaard, L. K., Astrup, T. F. 2017a. Material flow analysis of the Danish solid waste management system. Copenhagen.

Bisinella, V., Götze, R., Conradsen, K., Damgaard, A., Christensen, T.H., Astrup, T.F., 2017b. Importance of waste composition for life cycle assessment of waste management solutions. J. Cleaner Production 164, 1180–1191.

Boldrin, A., Hartling, K.R., Laugen, M., Christensen, T.H., 2010. Environmental inventory modelling of the use of compost and peat in growth media preparation. Resour. Conservation Recycling 12 (54). https://doi.org/10.1016/j.resconrec.2010.04.003.

Boldrin, A., Andersen, J.K., Christensen, T.H., 2011. Environmental assessment of garden waste management in the municipality of Aarhus Denmark. Waste Manage. 7 (31), 1560–1569.

Bourguignon, D., 2016. Closing the loop - New circular economy package. European Parliamentary Res. Service. https://doi.org/10.1111/j.1742-1241.2010.02575.x.

Cherubini, F., Peters, G.P., Berntsen, T., Strømman, A.H., Hertwich, E., 2011. CO_2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. GCB Bioenergy 3, 413–426. https://doi.org/10.1111/j.1757-1707.2011.01102.x.

Cherubini, F., Huijbregts, M., Kindermann, G., Van Zelm, R., Van Der Velde, M., Stadler, K., Strømman, A.H., 2016. Global spatially explicit CO_2 emission metrics for forest bioenergy. Sci. Rep. 6, 1–12. https://doi.org/10.1038/srep20186.

Cimpan, C., Maul, A., Jansen, M., Pretz, T., Wenzel, H., 2015a. Central sorting and recovery of MSW recyclable materials: a review of technological state-of-the-art, cases, practice and implications for materials recycling. J. Environ. Manage. 156, 181–199. https://doi.org/10.1016/j.jenvman.2015.03.025.

Cimpan, C., Rothmann, M., Hamelin, L., Wenzel, H., 2015b. Towards increased recycling of household waste: documenting cascading effects and material efficiency of commingled recyclables and biowaste collection. J. Environ. Manage. 157, 69–83. https://doi.org/10.1016/j.jenvman.2015.04.008.

Clavreul, J., Baumeister, H., Christensen, T.H., Damgaard, A., 2014. An environmental as- sessment system for environmental technologies. Environ. Model. Softw. 60, 18–30. https://doi.org/10.1016/j.envsoft.2014.06.007.

Dahlbo, H., Aalto, K., Eskelinen, H., Salmenperä, H., 2017. Increasing textile circulation—consequences and requirements. Sustain. Prod. Consum. 9, 44–57. https://doi.org/10.1016/j.spc.2016.06.005.

DEFRA, 2009. Municipal waste composition: A review of municipal waste component analyses. Available at. <http://randd.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=15133>.

Edjabou, M.E., Jensen, M.B., Götze, R., Pivnenko, K., Petersen, C., Scheutz, C., Astrup, T.F., 2015. Municipal solid waste composition: Sampling methodology, statistical analyses, and case study evaluation. Waste Manage. 36, 12–23. https://doi.org/10.1016/j.wasman.2014.11.009.

ecoinvent, 2018. Ecoinvent v 3.5. Available at: <https://v33.ecoquery.ecoinvent.org/Home/Index>.

Eriksen, M.K., Damgaard, A., Boldrin, A., Astrup, T.F., 2018. Quality assessment and circularity potential of recovery systems for household plastic waste. J. Ind. Ecol. https://doi.org/10.1111/jiec.12822.

European Commission-Joint Research Centre (EC-JRC), 2011. Institute for Environment and Sustainability: International Reference Life Cycle Data System (ILCD) Handbook- Recommendations for Life Cycle Impact Assessment in the European context. EUR 24571 EN. Luxemburg. Publications Office of the European Union.

EC (European Commission), 2016. EU reference scenario 2016. Energy, transport and GHG emissions – trends to 2050. <https://ec.europa.eu/energy/en/content/2013-%E2%80%93-eu-energy-transport-and-greenhouse-gas-emissions-trends-2050>.

EU (European Union), 2018. Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste. Available at: <https://eur-lex.europa.eu/eli/dir/2018/851/oj>.

EU (European Union), 2018. Directive (EU) 2018/852 of the European Parliament and of the Council of 30 May 2018 amending Directive 94/62/EC on packaging and packaging waste. Available at: <https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1529413058624&uri=CELEX:32018L0852>.

EC (European Commission), 2018c. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions – A European Strategy for Plastics in a Circular Economy. Commission staff working document, Brussels, p. 125.

Eurometaux, 2012. Recycling rates for metals. Available at: <https://www.eurometaux.eu/media/1510/electroniversionrecyclingratesdec2012_eurometauxeurofer.pdf>.

Faraca, G., Boldrin, A., Damgaard, A., Astrup, T.F., 2017. Challenges to a circular economy – the presence of impurities in wood waste for recycling. Proc, Sardinia, p. 2017.

Faraca, G., Tonini, D., Astrup, T.F., 2019. Dynamic accounting of greenhouse gas emissions from cascading utilisation of wood waste. Sci. Total Environ. 651, 2689–2700. https://doi.org/10.1016/j.scitotenv.2018.10.136.

Fruergaard, T., Christensen, T.H., Astrup, T., 2010. Energy recovery from waste incineration: assessing the importance of district heating networks. Waste Manag. 30, 1264–1272.

Haupt, M., Kägi, T., Hellweg, S., 2018. Modular life cycle assessment of municipal solid waste management. Waste Manag. 79, 815–827. https://doi.org/10.1016/j.wasman.2018.03.035.

Horodytska, O., Valdés, F.J., Fullana, A., 2018. Plastic flexible films waste management – a state of art review. Waste Manag. 77, 413–425. https://doi.org/10.1016/j.wasman.2018.04.023.

Intergovernmental Panel on Climate Change (IPCC), 2013. Climate Change 2013. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), The physical science basis. Working group I contribution to the fifth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, p. 1535.

ISO 14040, 2006. Environmental Management-Life Cycle Assessment-Principles and Framework. ISO, Geneva, Switzerland.

Krook, J., Mårtensson, A., Eklund, M., Libiseller, C., 2008. Swedish recovered wood waste : linking regulation and contamination. Waste Manag. 28, 638–648. https://doi.org/10.1016/j.wasman.2007.03.010.

Krook, J., Eklund, M., 2010. The strategic role of recycling centres for environmental performance of waste management systems. Appl. Ergon. 41, 362–367. https://doi.org/10.1016/j.apergo.2009.06.012.

Larsen, A.W., Merrild, H., Christensen, T.H., 2009. Recycling of glass: accounting of greenhouse gases and global warming contributions. Waste Manag. Res. 27, 754–762. https://doi.org/10.1177/0734242X09342148.

Miljøstyrelsen, 2017. Waste statistics 2015. Environmental project nr. 1955. Available at: <https://mst.dk/service/publikationer/publikationsarkiv/2018/sep/affaldsstatistik-2015/>.

Miljøstyrelsen, 2018. Affaldsstatistikken 2016. Miljøprojekt nr. 2020. Available at: <https://mst.dk/service/publikationer/publikationsarkiv/2018/sep/affaldsstatistik-2016/ (in Danish)>.

Muñoz, I. 2015. Example – Marginal Electricity in Denmark. Version: 2015-12-10. <www.consequential-lca.org>.

Møller, J., Jensen, M.B., Kromann, M., Neidel, T.L., Jakobsen, J.B. 2013. Miljø- og samfundsøkonomisk vurdering af muligheder for øget genanvendelse af papir,

pap, plast, metal og organisk affald fra dagrenovation (p.318, Appendix. 5.6 and 8). Miljøprojekt nr. 1458. (Environmental and socio-economic assessment of opportunities for increased recycling of paper, cardboard, plastic, metal and organic waste from municipal waste). In Danish

Natur, Miljø og Fødevareministeriet, 2018. Affaldsproduktion i Danmark fordelt på kilde og behandling (Waste production in Denmark by source and treatment). In Danish. Available at: (last accessed 17.10.2018). <https://www.ads.mst.dk/Forms/Reports/ReportsAffaldsproduktionOverview.aspx>.

Nilsson, P., Christensen, T.H. 2011. Waste collection: Systems and organisations. Chapter 6.2 in: Christensen, T.H. 2011. Solid waste technology and management. ISBN: 978-1-405-17517-3

Nørup, N., Pihl, K., Damgaard, A., Scheutz, C., 2018. Development and testing of a sorting and quality assessment method for textile waste. Waste Manag. 79, 8–21. https://doi.org/10.1016/j.wasman.2018.07.008.

Pivnenko, K., Eriksson, E., Astrup, T.F., 2015. Waste paper for recycling: overview and identification of potentially critical substances. Waste Manag. 45, 134–142. https://doi.org/10.1016/j.wasman.2015.02.028.

Reichel, A., De Schoenmakere, M., Gillabel, J., Martin, J., Hoogeveen, Y., 2016. Circular Economy in Europe - Developing the Knowledge Base. https://doi.org/10.2800/51444.

ResourceFutures, 2011. Final Report to Defra: WR1002 Detailed compositional assessment for municipal residual waste and recycling streams in England. Available at: <http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=17303>.

Rigamonti, L., Grosso, M., Møller, J., Sanchez, V.M., Magnani, S., Christensen, T.H., 2014. Environmental evaluation of plastic waste management scenarios. Resour. Conserv. Recycl. 85, 42–53. https://doi.org/10.1016/j.resconrec.2013.12.012.

Schmidt, A., Watson, D., Roos, S., Askham, C., Poulsen, P., 2016. Gaining benefits from discarded textiles - LCA of different treatment pathways. Denmark, Copenhagen.

Statistics Denmark, 2018. Waste generation by sector, kind of treatment and waste category. Available at: (last accessed 05.10.2018). <https://www.statbank.dk/AFFALD>.

Steffen, W., Richardson, K., Rockström, J., Cornell, S., Fetzer, I., Bennett, E., Sörlin, S., 2015. Planetary boundaries: guiding human development on a changing planet. Science 348 (6223), 1259855. https://doi.org/10.1126/science.aaa9629.

The Danish Government. (2013). Denmark without waste. Available at: <https://doi.org/978-87-03026-59-5>.

Tonini, D., Hamelin, L., Astrup, T.F., 2015. Environmental implications of the use of agro-industrial residues for biorefineries: Application of a deterministic model for indirect land-use changes. GCB Bioenergy 1–17. https://doi.org/10.1111/gcbb.12290.

Turner, D.A., Williams, I.D., Kemp, S., 2015. Greenhouse gas emission factors for recycling of source-segregated waste materials. Resour. Conserv. Recycl. 105, 186–197. https://doi.org/10.1016/j.resconrec.2015.10.026.

Villanueva, A., Eder, P., 2014. End-of-Waste Criteria for waste plastic for conversion. Luxemburg, JRC Technical Reports. 10.2791/13033.

Vis, M., Mantau, U., Allen, B., 2016. Study on the optimised cascading use of wood. No 394/PP/ENT/RCH/14/7689. Final report for the European Commission. Brussels 2016. 337 pages

Waste & Resources Action Programme (WRAP), 2012. Valuing our clothes: the evidence base. Available at: <www.wrap.org.uk/clothing>

Waste & Resources Action Programme (WRAP), 2016. Household Waste Recycling Centre (HWRC) Guide. Available at: <http://www.wrap.org.uk/sites/files/wrap/INH0449_HWRC_Guidance_2015_7%20FINAL.pdf>.

Weidema, B., Ekvall, T., Heijungs, R., 2009. Guidelines for application of deepened and broadened LCA.

Woolridge, A.C., Ward, G.D., Phillips, P.S., Collins, M., Gandy, S., 2006. Life cycle assessment for reuse/recycling of donated waste textiles compared to use of virgin material: An UK energy saving perspective. Resour. Conservation and Recycling 46, 94–103. https://doi.org/10.1016/j.resconrec.2005.06.006.